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# A RESOURCE MANAGER'S GUIDE FOR USING AQUATIC ORGANISMS TO ASSESS WATER QUALITY FOR EVALUATION OF CONTAMINANTS

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Fish and Wildlife Service

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# A RESOURCE MANAGER'S GUIDE FOR USING AQUATIC ORGANISMS TO ASSESS WATER QUALITY FOR EVALUATION OF CONTAMINANTS

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# PREFACE

The purpose of this report is threefold. First, it serves as a literature review regarding the use of aquatic organsims to assess ecological impacts. Second, it presents different methods to assess how data from biological collections have been traditionally treated regarding impact assessment. Third, it provides a scenario of how data could be collected and analyzed in a somewhat ideal situation.

This report is geared to managers and biologists unfamiliar with aquatic organisms and their use in impact analysis. This report is written primarily for resource managers that may be involved in contaminant-related problems in aquatic ecosystems. This is not a "how to" manual, but rather an overview of key literature relevant to using aquatic organisms to assess water quality for contaminants. For those who are more experienced with this topic, it will be evident that specialized areas of concern have been neglected.

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# INTRODUCTION

Contaminants in the aquatic environment are a serious and difficult issue confronting resource managers. Contaminants are toxic or otherwise harmful materials present in measurable concentrations (Lippmann and Schlesinger 1979) and usually fall into one of four categories: sediment, thermal, organic, and toxic chemicals.

It is difficult to measure the impact of a particular contaminant on a given aquatic ecosystem. Rand and Petrocelli (1985) list five factors that determine the vulnerability of an aquatic environment to a chemical insult: (1) physical and chemical properties of the chemical and its transformation products; (2) concentration of the chemical entering the ecosystem; (3) duration and type of inputs (acute or chronic, intermittent spill, or continuous discharge; (4) properties of the ecosystem that enable it to resist changes that could result from the presence of the chemical (e.g., pH buffering capacity of seawater) or that return it to its original state after the chemical is removed from the system (e.g., flushing of water from estuaries by tidal action); and (5) location of the ecosystem in relation to the release site of the chemical. Abiotic and biotic characteristics of a system also can modify the toxicity of a contaminant (Sprague 1985). Further problems regarding the source of the contaminant (point or nonpoint source) and synergistic interactions of groups of contaminants. These factors demonstrate the complexity of the contaminant problem and the difficulty in attempting to measure impacts on a given aquatic ecosystem.

Fish and wildlife resource managers need to be aware of the complexity of the problem, but are generally more concerned with documenting effects on biotic components of an aquatic ecosystem. Documentation can be accomplished through toxicity testing and field studies. Toxicity tests are used to evaluate the effects of a chemical under standardized and reproducible conditions that permit comparisons with other chemicals (Rand and Petrocelli 1985). Field studies attempt to document on-site changes at the community and species levels. Together, toxicity testing and field validation form the most powerful tools a resource manager can use in the assessment of contaminant impacts on aquatic ecosystems.

Biological indicators of pollution have an advantage over chemical measures, because biological indicators have a "pollution integrating" tendency (Ott 1978). Aquatic organisms tend to respond to the entire historical record of water quality. Occasional toxic perturbations in an aquatic ecosystem can go undetected in routine water quality monitoring activities. The presence of these pollutants, however, would still be indicated by the types and numbers of organisms present.

This report reviews pertinent literature concerning the potential use of aquatic organisms as indicators of aquatic contaminants. The first section reviews toxic effects, followed by a section on taxonomic groups that have been used to indicate water quality. Next, we give references for field sampling and discuss statistical methods for data analysis. The final section provides a scenario of a toxic spill into a western river and the actions that could be taken.

Because of this topic's broad range, emphasis is placed on freshwater systems. Fish have been excluded from this report due to the large amount of information available (Sprague 1969; Morgan 1975; Ott 1978; Rand and Petrocelli 1985).

## TOXIC EFFECTS

Toxic effects fall into two categories, acute and chronic. Acute effects occur rapidly as a result of short-term exposure to a chemical, usually within a few hours, days, or weeks (Rand and Petrocelli 1985). Acute effects are generally severe. The most common acute effect is lethality or mortality. Rand and Petrocelli (1985) state that a chemical is considered acutely toxic if it kills 50% or more of the exposed population of test organisms in a relatively short period of time, such as 4 to 14 days.

Chronic toxic effects occur when the toxicant induces deleterious effects as a result of either a single exposure or long-term (repeated) exposures (Rand and Petrocelli 1985). Rand and Petrocelli (1985) also noted that often there may be a long latency period before the consequences of these effects become evident, especially if the concentration exposure is low. Chronic toxic effects may be lethal or sublethal. The most common lethal effect is the failure of the chronically or subchronically exposed organism to produce viable offspring (Rand and Petrocelli 1985). Typically, sublethal effects are behavioral (decrease in the ability to swim, attraction-avoidance, and predator-prey relationships), physiological (resistance to disease, growth, reproductive capacity, development), biochemical (blood enzyme and ion levels), or histological (APHA 1980; Rand and Petrocelli 1985). However, some sublethal effects can indirectly result in lethalities. For example, behavioral changes, or loss of the olfactory sense, may reduce the ability of the aquatic organism to escape from predators or locate food, thereby leading to death. Other sublethal effects may have little or no effect because they are rapidly reversible, diminish, or cease with time (Rand and Petrocelli 1985).

If a resource manager needs information regarding the acute, chronic, or subchronic toxic effects of a specific chemical compound, there are several sources of available. One source is the U.S. Environmental Protection Agency [Office of Toxic Substances, 401 M St., Washington, DC 20460, (202) 426-7010, or Biological Methods Branch, Envir. Mon. and Support Lab., Cincinnati, OH 45268, (513) 569-7337], which has numerous publications on specific compounds. Another source is the Handbook of Acute Toxicity of Chemicals to Fish and Aquatic Invertebrates by W.W. Johnson and M.T. Finley, published by the U.S. Fish and Wildlife Service (Resource Publication 137). Additional information can be obtained from the annual literature reviews published in the Journal of Water Pollution Control Federation.

## A LITERATURE REVIEW OF AQUATIC TAXONOMIC GROUPS THAT HAVE BEEN USED TO EVALUATE IMPACTS OF CONTAMINANTS

### Bacteria

Bott (1973) states that "bacteria are important in the assessment of water quality because certain organisms may be used as indicators of pollution of various types, and because bacteria possess diverse metabolic capabilities that bring about mineralization of substances, many of which other organisms are unable to use."

Coliform bacteria are one of the most widely used indicators of fecal pollution and potential pathogens in water. Coliform counts are inexpensive and are done by State laboratories, public health departments, and private laboratories. Results of such tests can be useful to resource managers when urine and feces from man, domestic livestock, or wildlife contaminate water from storm sewer runoff and runoff from natural areas (Bott 1973).

Coliform bacteria are an indicator of disease-causing microorganisms because coliform bacteria are usually found in large numbers where these pathogens exist (Kimmel and Moon 1982). Examples of pathogens most often associated with outbreaks of waterborne diseases include *Salmonella* spp., which causes typhoid fever, *Pasturella* spp., which causes tularemia (rabbit fever), and *Vibrio* spp., which causes cholera (Bott 1973).

Bacteria have been shown to be indicators of nonfecal pollution. Sulfate and nitrate reducers, phenol decomposers, and hydrocarbon oxidizers have been found in higher numbers down river from cities (and industrial discharge) than up river (Luchterowa 1966; Luchterowa and Grela 1969). *Thermus aquaticus*, an extreme thermophile, has been found only where heated effluents were introduced in a thermally polluted stream and several coldwater springs (Brock and Yoder 1971). Brock and Yoder (1971) suggested that *Thermus aquaticus* might be useful as an indicator of thermal pollution. Bott (1973) states the following: "Considerable attention has been focused on nuisance growth of *Sphaerotilus* spp. in streams. Large populations of this bacterium and associated organisms chracteristically develop under conditions of high organic loading, such as the introduction of raw sewage or improperly stabilized pulp mill effluents. The presence of this 'sewage fungus' leads to increased oxygen consumption and the destruction of habitats, as well as eggs and larval organisms by coating them. In addition to bringing about unsightly conditions, growths may foul lines and clog nets, thus interefering with fishing activities."

### Algae

Algae provide resource managers with one of the best tools for evaluating impacts of contaminants in marine, lentic, and lotic aquatic systems. In marine and lentic conditions, phytoplankton (free floating) communities are usually examined, whereas in lotic systems, periphyton (attached forms) communities are of interest.

Algal species and communities are sensitive to environmental changes, and growth may be either stimulated or inhibited by the presence of various chemicals (Rand and Petrocelli 1985). Patrick (1973) noted that if someone wanted to continually monitor a body of water to determine if anything was unfavorable to aquatic life, the best way was to use algal community assays. Patrick (1973) stated that "algal cells integrate all stresses that might deleteriously affect growth and reproduction or, in a similar way, they integrate all stimulants that may be present." Monitoring artificial substrates (algal cells) is more useful for continually monitoring the condition of the aquatic environment than chemical analysis (Patrick et al. 1954). Chemical analysis only defines amounts of chemicals present at a given time. Patrick (1973) recommends the use of both chemical analysis and bioassay to more specifically determine the cause of shifts in community structure.

Phytoplankton have long been used to indicate water quality, because some species flourish in eutrophic conditions and others are sensitive to contaminants (APHA 1980). Species that indicate clean water include Melosira islandica, Cyclotella ocellata, and Dinobryon spp. Species that indicate contaminated water include Nitzschia palea, Microcystis aeruginosa, and Aphanizomenon flos-aquae (APHA 1980).

Examples of field studies that have documented impacts of contaminants on algae in natural lakes and streams and in artificial streams and ponds include Besch et al. (1972); Dickman and Gochnauer (1978); Roberts et al. (1979); Squires et al. (1979); de Noyelles et al. (1984); Stevenson and Stoermer (1982); Kosinski (1984); and Kosinski and Merkle (1984).

Bioassay studies on algae also are important sources of information for resource managers. For example, Peterson et al. (1984) found that metal toxicity to algae was a highly pH-dependent phenomenon. A resource manager can incorporate this information into field studies on impacts of heavy metals by monitoring pH as a variable.

Schelske (1984) provides a good review of phytoplankton bioassay techniques for freshwater systems, and Maestrini et al. (1984) review techniques for seawater. One approach for monitoring coastal waters is the use of macrophytic algae as indicators of contaminants (Levine 1984).

## Macrophytes

Macrophytes can be used as indicators of heavy metals and eutrophication in lakes. Seddon (1972) studied 120 lakes in a 6-year period and found that trophic level condition, defined by conductivity and water hardness, explained the distribution of macrophytes. Macrophytes were classified into three groups: (1) species tolerant to all conditions, (2) species confined to eutrophic conditions and tolerant to impoverished conditions, and (3) those found in impoverished conditions (Seddon 1972).

Jana and Choudhuri (1984), in a laboratory study, found that species of aquatic submerged plants exposed to heavy metals (Hg, Pb, Cd, and Cu) became senescent. The senescent condition was attributed to inhibition of nucleic acid biosynthesis, impaired protein synthesis, and altered membrane integrity of tissue, all caused by heavy metals. This study also documented synergistic effects of contaminants.

Overall, little information exists on using macrophytes as indicators of contamination.

## Protozoans

Protozoan communities have not been used widely in assessing impacts of contaminants in lakes, but may be a useful tool. Hart and Cairns (1984) found that protozoan communities from eutrophic conditions were less diverse, and that a greater proportion of species was tolerant to low concentrations of copper than in oligotrophic conditions. Hart and Cairns (1984) explained that lower diversity was caused by a greater proportion of species in high numbers, which may give eutrophic protozoan communities a greater chance of surviving toxic stress; species with high numbers have a greater chance of a few individuals surviving, which in turn would preserve species richness (number of species).

## Zooplankton

The zooplankton in freshwater systems is dominated by three major groups: rotifers cladocerans, and copepods (Wetzel 1975). Like phytoplankton, zooplankton are useful indicators of water quality because their short life cycles allow them to respond quickly to environmental change (APHA 1980).

The use of phytoplankton and zooplankton as indicators of water quality may be limited due to their transient nature and patchy distribution (APHA 1980). For this reason, they are not as reliable as other taxonomic groups for determining impacts of contaminants and will not be discussed further. Additional information concerning zooplankton dynamics in response to trophic level changes can be found in Wetzel (1975), and ecological information can be obtained from Kerfoot (1980).

## Macroinvertebrates

Benthic macroinvertebrates are animals that inhabit the substratum of lakes and streams and are visible to the naked eye (Isom 1978). Organisms that are considered benthic macroinvertebrates are listed in Table 1 (APHA 1980).

Table 1. Common names of animals that are considered as macro-invertebrates by APHA (1980).

| Phylum | Class | Order | Common name |
|---|---|---|---|
| Porifera | | | Sponges |
| Platyhelminthes | Turbelaria | | Flat worms |
| Nematoda | | | Round worms |
| Mollusca | | | Bivalves |
| | | | Snails |
| Annelida | | | Segmented worms |
| | Hirudinea | | Leeches |
| Anthropoda | Merostomata | | Horseshoe crab |
| | Arachnida | Acari | Watermites |
| | Crustacea | Wysidea | Shrimp |
| | | Decapoda | Crayfish |
| | | | Crabs |
| | | Isopoda | Sowbugs |
| | Insecta | Collembola | Springtail |
| | | Plecoptera | Stoneflies |
| | | Ephemeroptera | Mayflies |
| | | Coleoptera | Beetles |
| | | Zygoptera | Dragonflies |
| | | Anisoptera | Damselflies |
| | | Megaloptera | Hellgrammites |
| | | Neuroptera | Spongilla flies |
| | | Trichoptera | Caddisflies |
| | | Hemipterans | True bugs |
| | | Dipterans | Flies |
| Bryozoa | | | Moss animals |
| Cnidaria | Hydrozoa | | Hydras |
| | Anthozoa | | Sea anemones |
| | Scyphozoa | | Jellyfish |
| | Anthozoa | | Corrals |
| Mollusca | Bivalvia | | Bivalves |
| | Gastropoda | | Snails |
| Echinodermata | Asteroidea | | Star fishes |
| Chordata | | | Sea pork |
| | | | Sea squirts |

APHA (1980) states the following regarding the use of macroinvertebrates as biomonitors:

> The composition and density of macroinvertebrate communities in streams and lakes are reasonably stable from year to year in unperturbed environments. However, seasonal fluctuations associated with life-cycle dynamics of individual species may result in extreme variation at specific sites within any calendar year. Most aquatic habitats (particularly free-flowing streams) with acceptable water quality and substrate conditions support diverse macroinvertebrate communities in which there is a reasonably balanced distribution of species among the total number of individuals present. Such communities respond to changing habitat quality by adjustments in community structure. However, many estuaries are dominated by a few species. Small changes in their relative numbers may not be indicative of changes in water quality.
>
> Macroinvertebrate community responses to environmental perturbations are useful in assessing the impact of municipal, industrial, oil, and agricultural wastes, as well as impacts from other land uses on natural water bodies. Two situations for which the patterns of macroinvertebrate community structure change have been documented are organic loading and toxic chemical pollution. Severe organic pollution usually results in a restriction in the variety of macroinvertebrates to only the most tolerant ones and a corresponding increase in density of those tolerating the polluted conditions, usually low dissolved oxygen. On the other hand, pollution by toxic chemicals may eliminate the entire macroinvertebrate community from an affected area. Not all cases conform to these described, because conditions may be mediated by other environmental and biological factors.

Lenat et al. (1980) provide a good review of studies on macroinvertebrates regarding the impacts of three categories of contaminants: sediment, temperature, and toxic materials.

Studies that have been documented that increased sediment decreases density to a greater degree than species richness include: Tebo (1955), Cordone and Kelley (1960), Chutter (1969), Gammon (1970), Pearson and Jones (1975), and Rosenberg and Snow (1975).

Studies on temperature show two major trends. The first is that moderate temperature increases rarely affect the benthos (Markowski 1970; Langford and Aston 1972). The second is that temperatures in excess of 30 °C result in substantial changes in the benthic macroinvertebrate community (Gallup et al. 1975, in Lenat 1978).

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Benthic community responses to toxic materials and organic wastes are markedly different (Lenat et al. 1980). Hocutt (1975) illustrated the difference by the ordination technique. Species richness and density are both depressed by severe doses of toxic materials. The remaining tolerant species often may be characteristic of a particular stress. Limited doses of a toxic material may have variable effects on density, e.g., density may show a decrease compared to control sites when predatory intensity is increased. This type of reaction has been observed for pesticides (Cape 1966; Muirhead-Thompson 1971; Holden 1972), and acid mine drainage (Hall 1951; Roback and Richardson 1969; Simmons 1972; Dill and Rodgers 1974).

Some metals also produce a reaction typical of toxic materials (Lenat et al. 1980). Acute toxic responses have been observed in macroinvertebrates when metal concentrations are elevated (Warnick and Bell 1969); however, when metal concentrations are low, lethal and sublethal responses have been observed (Spehar et al. 1978). Shifts from chironmids to oligochaetes have been observed in a lake contaminated with cadmium, chromium, and zinc (Wetzel et al. 1977). Other studies have demonstrated shifts in dominant species and a decrease in species richness when levels of copper, lead, and zinc were elevated (Butcher 1955; Jones 1958; Winner et al. 1975).

Benthic community changes resulting from trace metal inputs have not been well documented (Lenat et al. 1980). Shuman et al. (1977) found that whole body concentrations of trace materials in macroinvertebrates indicated the trace metal concentration in the surrounding water and sediment; therefore, they concluded that the macroinvertebrates may be useful as indicators of long-term metal inputs. However, Smock (1979) felt that the usefulness of macroinvertebrates as indicators of short-term, pulse inputs of metals, was complicated by the rapid loss rates of absorbed metals, and because the major proportion of trace metals was associated with gut contents.

Other studies using macroinvertebrates have shown contaminant effects. A good example of study design and use of community structure to document impacts of toxic metals on streams is given by LaPoint et al. (1984). Winner et al. (1980) working on two second-order streams in Ohio found that all macroinvertebrates, with the exception of chironomids (midges) and tubificids (aquatic earthworms), were eliminated from stream sections with high heavy metal contamination. Because of a direct relationship between chironomid proportions in samples and heavy metal contaminants, Winner et al. (1980) suggest that chironomids can serve as a good indicator of heavy metal contamination.

Studies on subgroups of benthic macroinvertebrates also have been useful in documenting effects of contaminants. Gastropods have been used as indicators of trophic levels in lakes (Clarke 1979a) and in artificial streams to document changes in populations of two species exposed to sewage effluent (Watton and Hawkes 1984). Mollusks of the family Sphaeridae have been used to document trophic stages in lakes (Clarke 1979b). Milbrink (1973, 1983) and Howmiller and Scott (1977) have shown that oligochaetes are good indicators of contaminants.

Aquatic insects also have been useful indicators of contaminants. Cushman (1984) found that chironomid deformities of the mentum (a mouth part) served as a good indicator of contamination from synthetic-coal-derived oil. Wilson and McGill (1977) proposed using chironomid pupal exuviae to monitor water quality and used sewage effluent as a practical example. Pupal exuviae refers to the discarded skin of an insect after it has molted into an adult form (Merrit and Cummins 1978).

One noteworthy approach concerning the use of aquatic insects as indicators of contaminants comes from Petersen and Petersen (1985). They used net anomalies of netspinning caddisflies (Hydropsyche) as indicators of heavy metal wastes and pulp mill effluents. Petersen and Petersen (1985) state that the frequency and type of net anomaly is related to the concentration and type of toxic substance, which allows the direct translation of laboratory results to field observations.

No discussion of macroinvertebrates is complete without discussing the phenomenon of drift. Drift refers broadly to the downstream transplant by current of benthic animals normally living on or in bottom substrates (Wiley and Kohler 1984). Drift has been classified into three categories: catastrophic (pulsed), behavioral (periodic), and constant (continuous) (Waters 1972). Wiley and Kohler (1984) believed these terms only distinguished between the temporal properties that different types of drift exhibited. There are two modes of drift entry into the water column, active and passive (Wiley and Kohler 1984). Active entry into the water column by voluntary abandonment of the substrate is a common behavioral response of aquatic insects to stress stimuli. Passive entry into the water column is either accidental or unintentional release and subsequent erosion. Erosion is defined as, given a sufficient current, any insect that loses its hold on the substrate and is swept immediately into the water column (Wiley and Kohler 1984). Eidt and Weaver (1983) determined threshold concentrations of aminocarb (used for spruce budworm) that would cause drift of stream insects. Dermott and Spence (1984) found increases in drift rates as a response to lampricide treatment. Back et al. (1983) found that temephos (an organophosphorus insecticide), when applied to bogs in subarctic Quebec for mosquito control, increased lotic insect drift in small streams draining bogs. The stonefly, Podmosta macdunnoughi, was particularly sensitive to temephos (Back et al. 1983).

## Amphibians

The amphibians are another group that has not been extensively used for monitoring impacts of contaminants on aquatic systems. Birge et al. (1981) used frogs in conducting embryo-larval toxicity tests. Birge and Black (1981) also used these tests to conduct on-site biomonitoring of municipal and industrial effluent; they concluded that the bullfrog (Rana catesbiana) and leopard frog (Rana pipians) were preferred species to fish and other amphibians for toxicity testing. One important feature of using amphibians in contaminant work is that a great deal of information is available regarding their embryologic development. Thus, amphibians may be a useful group in determining modes of action of contaminants on mechanisms of development.

## Considerations for Selecting a Taxonomic Group for Contaminants Work

Selection of a taxonomic group for contaminant effects studies is largely dependent on the resources available and taxonomic experience of personnel. For example, to properly collect and analyze plankton data requires sampling equipment (including a boat), phase-contrast compound microscopes, vacuum flasks and millipore filters for concentrating samples, mounting mediums, counting slides, and taxonomic keys. Another consideration is the volume of samples to be protected.

The uses, materials, experience, and relative cost of the taxonomic groups previously discussed, with the exception of amphibians, are listed in Table 2. Amphibians were excluded because they are just beginning to be used to evaluate contaminant effects. Cost becomes difficult to assess, because it is dependent on the facilities and personnel available. The column of ranked costs in Table 2 was determined by talking to consultants and taxonomists. Furthermore, the ranks are based on the assumption that the resource manager has no experience with taxonomy. Once again, it is the facilities and personnel available that may be a consideration in choosing a group. For example, a consulting firm with a zooplankton taxonomist may find it is cheaper to use zooplankton than macroinvertebrates. The quality of personnel is extremely important because once most of the species of a sample have been identified, sample processing is greatly accelerated. However, the selection of a taxonomic group is highly dependent on the nature of the contaminant. For example, bacteria may be the best group to use when considering a fecal contamination problem. Therefore, taxonomic groups should be chosen first on their ability to document an effect, and second, by what facilities and personnel are available.

In determining the relative costs in Table 2, several considerations were taken into account. Algae require more sample preparation than the other groups, because diatoms have one preparation and nondiatom species another. It is harder to find personnel experienced in algal and zooplankton taxonomy than aquatic insect identification. Generally, macroinvertebrates are easier to sample than algae or zooplankton. Macrophytes are easy to sample, but little is known regarding their response to contaminants; however, maps of their distribution may be valuable in documenting long-term changes due to a contaminant. Vegetation mapping is relatively cheaper than sampling for macroinvertebrates. Finally, bacteria were determined the least expensive group to use because all that is needed is a water sample, which outside laboratories can process for a relatively low cost.

Table 2. Uses, materials needed, experience, and relative cost for the taxonomic groups discussed in the text. Relative cost is ranked from 1 to 5, with 1 representing the least expensive and 5 the most expensive. For determination of cost consult text.

| Group | Uses | Materials | Experience | Relative cost |
|---|---|---|---|---|
| Bacteria | Fecal contamination<br>Thermal pollution<br>Toxic compounds<br>Organic enrichment | Laboratory | Bacteria techniques | 1 |
| Algae | Nutrients<br>Thermal pollution<br>Sedimentation<br>Toxic compounds | Microscopes<br>Collecting gear<br>Sample prep<br>Counting slides | Taxonomy<br>Field collection<br>Lab prep | 5 |
| Macrophytes | Nutrients<br>Thermal pollution<br>Sedimentation<br>Toxic compounds | Collecting gear<br>Plant press<br>Stero scope | Taxonomy<br>Field collection<br>Specimen mounts | 2 |
| Zooplankton | Nutrients<br>Thermal pollution<br>Sedimentation<br>Toxic compounds | Microscopes<br>Collecting gear<br>Sample prep | Taxonomy<br>Field collection<br>Lab prep | 4 |
| Macroinvertebrates | Nutrients<br>Thermal pollution<br>Sedimentation<br>Toxic compounds<br>Organic enrichment | Stereo scope<br>Collecting gear | Taxonomy<br>Field collection<br>Sample prep | 3 |

## A REVIEW OF DATA COLLECTION TECHNIQUES AND METHODS FOR TREATING FIELD DATA

### Sampling Procedures and Field Collection Techniques

The results of a study are only as good as the data, which in turn are highly dependent on sampling design and techniques. It is not our intention to give detailed explanations of sampling, because the cited sources cover this material in clear concise fashion.

As an overall guide to field collection techniques for aquatic organisms consult APHA (1980), which is published jointly by the American Public Health Association, American Water Works Association, and the Water Pollution Control Federation. For information concerning sampling design for stream systems, consult Armour and Platts (1983); for lakes consult Lind (1985). Other sources of information for sampling design are Elliott (1977), Southwood (1978), and Green (1979).

In field monitoring of populations and communities of aquatic organisms impacted by contaminants, the general strategy is to sample perturbed sites and compare them to samples from adjacent (unaffected) sites. In essence, unaffected sites serve as controls.

## Treatment of Data

This section will focus on indices that can be used to make community comparisons between different sampling stations to determine impacts of contaminants.

## Biotic Indices

Biotic indices attempt to combine known tolerance data for organisms with qualitative measures of their abundance to determine water quality (Lenat et al. 1980). Lenat et al. (1980) describe three steps in the development of a biotic index: (1) compile a list of organisms found or expected to be found, (2) assign quality factors (tolerance factors) to each taxon on the list, and (3) calculate a Trent index on the actual data.

This report focuses on four biotic indices: the Trent Index (Woodiwiss 1964), the Chutters Biotic Index (Chutter 1969), the Chandler Biotic Score (Chandler 1970), and the Average Chandler Biotic Score (Ballogh et al. 1976).

The Trent Index was developed in England and uses types and groups of taxa present to determine a score between 0 and 10, with 10 representing clean water and 0 heavily polluted water. Ballogh et al. (1976) state that this index is generally based on the order in which benthic macroinvertebrate species disappear with deteriorations in water quality. Table 3 displays the Trent Index system and group categories.

To show how this system works, we will use an example. Suppose a sample was collected with eight groups, including two species of plecopteran nymphs, three species of ephemeropteran nymphs, and three species of trichopteran nymphs. The investigator would go to Table 3 and look in the column of 6-10 for the total number of groups present and then look in the row designated plecopteran nymphs present (more than 1) to find the biotic score of 8. The value of 8 would indicate a stream of rather high water quality.

A modified version of the Trent Index for use in Denmark was devised by Andersen et al. (1984). The major changes are: (1) using pollution indicator species as negative taxa, which lowers the index value; and (2) replacing the taxonomic groups used in Table 3 with negative and positive index groups, where the number of negative groups is subtracted from positive to determine the biotic score.

Table 3. The Trent Index as given by Ballogh et al. (1976).

| Taxonomic group | Criteria | Total number of groups present<br>0-1 | <br>2-5 | Biotic index<br>6-10 | <br>11-15 | <br>16 + |
|---|---|---|---|---|---|---|
| Plecoptera nymphs present | More than one species | - | VII | VIII | IX | X |
| | One species only | - | VI | VII | VIII | IX |
| Ephemeroptera nymphs present | More than one species* | - | VI | VII | VIII | IX |
| | One species only* | - | V | VI | VII | VIII |
| Trichoptera larvae present | More than one species | - | V | VI | VII | VIII |
| | One species only+ | IV | IV | V | VI | VII |
| *Gammarus* present | All above species absent | III | IV | V | VI | VII |
| *Asellus* present | All above species absent | II | III | IV | V | VI |
| Tubificid worms and/or red chironomid larvae present | All above species absent | I | II | III | IV | - |
| All above types absent | Some organisms, such as *Eristalis tenax*, not requiring dissolved oxygen may be present | 0 | I | II | - | - |

GROUPS

The term "Group" here denotes the limit of identifcation that can be reached without resorting to lengthy techniques. Thus, the Groups are as follow:

Each known species of Platyhelminthes (flatworms).
Annelida (worms) excluding genus *Nais*.
Genus *Nais* (worms).
Each known species of Hirudinae (leeches).
Each known species of Mollusca (snails).
Each known species of Crustacea (log louse, shrimps).
Each known species of Plecoptera (stonefly).
Each known species of Ephemeroptera (mayfly) excluding *Bactis rhodani*.
*Bactis rhodani* (mayfly).
Each family of Trichoptera (caddisfly).
Each species of Neuroptera (caddisfly).
Each species of Neuroptera larvae (alderfly).
Family Chironomidae (midge larvae )except *Chironomous Ch. thummi*.
Chironomous Ch. *thummi* (blood worms).
Family Simulidae (black fly larvae).
Each known species of other fly larvae.
Each known species of Coleoptera (beetles and beetle larvae).
Each known species of Hydracarina (water mites).

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+ *Bactis rhodani* excluded
* *Bactis rhodani* (Ephem.) is counted in this section for the purpose of classification.

The Chandler Biotic Score uses a system similar to the Trent Index, but incorporates a more detailed list of macroinvertebrates and includes data regarding the abundance of each taxon in calculating a score. Table 4 displays the information necessary to calculate the Chandler Biotic Score for a sample. Once again, an example calculation will help clarify how this index works. Suppose 25 Gammarus were found in a sample. The investigator would go to Table 4a to determine that Gammarus are considered common. Next, he would go to Table 4b and look for the row labeled Gammarus and move to column "C" for common to find the score of 40. Similar values would be calculated for all other taxa in the sample. All the scores for each taxa are then summed to arrive at the Chandler Biotic Score. There is no upper limit to the Chandler Biotic Score, but the higher the score, the higher the water quality. Also, because the score is in continuous gradations from high to low, it is easier to assess borderline conditions.

The Average Chandler Biotic Score (Ballogh et al. 1976) is computed in the same manner as the Chandler Biotic Score, except that the Chandler Biotic Score is divided by the number of species, species groups, or genera used in the original computation. The advantage of using the Average Chandler Biotic Score is that it is not dependent on the number of species taken in the sample, which is a common source of error (Murphy 1978).

Table 4(a). The Chandler Biotic Score with 4(a) giving relative abundance levels, and 4(b) giving score system for groups.

| Sample abundance levels | |
|---|---|
| Level | Number of individuals per 5-min sample |
| Present | 1 to 2 |
| Few | 3 to 10 |
| Common | 11 to 50 |
| Abundant | 51 to 100 |
| Very abundant | more than 100 |

### Table 4(b). Chandler Biotic Score system.

| | Groups present in sample | Increasing abundance<br>Points scored | | | | |
|---|---|---|---|---|---|---|
| | | P | F | C | A | V |
| Each species of: | *Planaria alpine*, Taenopterygidae, Periidae, Periodidae, Isoperlidae, Chloropedidae | 90 | 94 | 98 | 99 | 100 |
| Each species of: | Leuctridae, Capniidae, Nemouridae (excluding *Amphinemura*) | 84 | 89 | 94 | 97 | 98 |
| Each species of: | Ephemerotera (excluding *Baetis*) | 79 | 84 | 90 | 94 | 97 |
| | Cased caddis, Megaloptera | 75 | 84 | 90 | 94 | 97 |
| | *Ancylus* | 70 | 75 | 82 | 87 | 91 |
| | *Rhyacophila* (Trichoptera) | 65 | 70 | 77 | 83 | 88 |
| Genus of: | *Dicranota*, Limnophera | 60 | 65 | 62 | 78 | 84 |
| | *Simulium* | 56 | 61 | 67 | 73 | 75 |
| | Coleptera, Menatoda | 51 | 55 | 61 | 66 | 72 |
| | *Amphinemura* (Plecoptera) | 47 | 50 | 54 | 58 | 63 |
| | *Baetis* (Ephemeroptera) | 44 | 46 | 48 | 50 | 52 |
| | *Gammarus* | 40 | 40 | 40 | 40 | 40 |
| Each species of: | Uncased caddis (excluding *Rhyscphila*) | 38 | 36 | 35 | 33 | 31 |
| | Tricladida (excluding *P. alpina*) | 35 | 33 | 31 | 29 | 25 |
| Genus of: | *Hydracarina* | 32 | 30 | 28 | 25 | 21 |
| Each species of: | Mollusca (excluding *Ancylus*) | 30 | 28 | 25 | 22 | 18 |
| | Chironomids (excluding *C. riparius*) | 28 | 25 | 21 | 18 | 15 |
| Each species of: | *Glossiphonia* | 26 | 23 | 20 | 16 | 13 |
| | *Asellus* | 25 | 22 | 18 | 14 | 10 |
| | Leech (excluding Glossiphonia, Haemopsis) | 24 | 20 | 16 | 12 | 9 |
| | *Haemopsis* | 23 | 19 | 15 | 10 | 7 |
| | *Tubifex* sp. | 22 | 18 | 13 | 12 | 9 |
| | *Chironomus riparius* | 21 | 17 | 12 | 7 | 4 |
| | *Nais* | 20 | 16 | 10 | 6 | 2 |
| Each species of: | Air breathing species | 19 | 15 | 9 | 5 | 1 |
| | Number of animal life | | | 0 | | |

Chutter (1972) found that the Trent Index was not applicable to South African streams and proposed an alternative index that incorporates quantitative data and uses tolerance factors scaled from 1 to 10 in the following formula:

$$BI = \frac{\sum_{i=1}^{k} n_i \, a_i}{N}$$

where BI = Biotic Index

i = the ith taxon

k = the number of taxa

n = density of the ith taxon

a = quality or tolerance factor for the ith taxon

N = the total number of individuals from all taxa

Chutter's scale goes from 0 to 10, with 0 representing clean water and 10 heavily polluted water. For further information regarding this index consult Chutter (1972).

Biotic indices may be useful in monitoring change between sampling stations to monitor impacts of contaminants. The benefits and limitations of using biotic indices are given in Table 5. One major problem with biotic indices is their geographic limitations for lists of organisms and the determination of tolerance factors. Lenat et al. (1980) suggested tolerance factors be determined using referee boards of ecologists, rather than one individual. An example of such a system is the North Carolina Division of Environmental Management Analysis System described by Penrose and Lenat (1978).

## Diversity Indices

Species diversity incorporates both species number (richness) and relative abundance in a community, and is used as a more representative measure of community structure rather than simple species number indices (Giller 1984). Typically, when a fauna (or a flora) is sampled, a few species are represented by a lot of individuals, and a large number of species are represented by few individuals (Southwood 1978).

Table 5. Chart summarizing the pros and cons of using biotic indices discussed in the text (after Ballogh et al. 1976).

| Biotic index | Pros | Cons |
|---|---|---|
| Trent Index | Classifies polluted waters; relatively easy sampling; identification of species reduced by selection of key species; uses a simple linear scale of index values; index values understandable to nonbiologists. | Fixed-level index makes borderline cases difficult; index has narrow range; lacks range in clean to moderately polluted waters; no quantitative data used; only valid for Trent River; drift organisms can have large impact on results; loss of information from grouping of organisms; usually not responsive to heavy metals. |
| Chandler Biotic Score | Gradation of score values allows a wider range of water qualities to be classified; includes a more comprehensive list of organisms arranged by tolerance to organic pollution; relative abundance data is included in analysis; can detect change between clean and mildly polluted systems; no rigorous demands on sampling procedures. | List of organisms and their tolerance to organic pollution is subject to debate; requires more time to process samples than Trent Index; scale is harder for non-biologists to understand. |
| Average Chandler Biotic Score | Same as Chandler Biotic Score, except it is not dependent on the number of species taken in a sample. | Same as Chandler Biotic Score. |
| Chutter Score | Same as Trent Index, except quantitative data incorporated into the index. | Same as Trent Index, except more data has to be counted. |

An index of species diversity is a single number, an index, that combines information on both the number of species present in a community and on the relative abundance (evenness) with which individual organisms are distributed among species (Pielou 1975). Diversity indices are often used to summarize or reduce abundance data to a single biological variable for each sample (Green 1975). Patrick (1949) justified using species diversity as a measure of environmental conditions. She noted that in streams with healthy conditions a great many species representing various taxonomic groups were present. In streams with unhealthy conditions (a toxicant introduced) many species would be eliminated, decreasing competition among the remaining species, and allowing them to increase in number.

The use of diversity measures has recieved criticism (Lenat et al. 1980), the most valid of which is the loss of information regarding species composition (James and Rathbun 1981). Thus, two communities may have the same diversity values, but different species composition. Green (1979) believes diversity indices have been extensively and often uncritically applied without regard to the assumptions implicit in the various diversity forumlae and with too little emphasis placed on the critical importance of proper sampling design. Green (1979) strongly emphasizes being able to concisely state the question you are asking and deemphasizes the use of diversity indices.

Kaesler et al. (1978) stated that depending on one's point of view, not all the characteristics of diversity indices are necessarily disadvantageous. Pearson et al. (1967) defend the use of diversity indices. Pearson et al. (1967) believe that diversity indices help ecology to progress, which has been hampered by the lack of quantitative analytical techniques used for consolidating long taxonomic lists, and they can be reviewed and understood by nonecologists (Kaesler et al. 1978). Lastly, Kaesler et al. (1978) stated "Our further investigation of diversity indices is aimed at making them more efficient tools, while stressing their limitations so that all users of the results of environmental surveys need not be taxonomists, ecologists, and natural historians."

As a tool to measure contaminant impacts, diversity indices apply when monitoring organic wastes, which follow a pattern of increased species richness and decreased density (Lenat et al. 1980). Winget and Mangum (1979) report that contaminants added in small concentrations may increase diversity estimaters for the following reasons.

(1) When the toxin being added eliminates one or more of the very fragile species, these may be replaced by other more hardy species. These more hardy species may be able to account for a higher percentage of the total numbers than the replaced fragile forms, thus spreading the total population over a larger number of species and increasing community diversity.

(2) When the perturbation is nonselective it has a greater impact on the dominant forms simply because there are more of them exposed, causing a reduction in their numbers. This in itself would even out the distribution of numbers over total species and thus result in an increase in community diversity.

(3) The addition of a substance may provide a previously limited chemical, allowing additional species to become established, thus increasing community diversity.

Several diversity measures have been proposed by numerous authors: Gleason (1922), Shannon and Weaver (1949), Simpson (1949), Margalef (1951, 1957), Brillouin (1962), Menhinick (1964), Pielou (1966, 1975), McIntosh (1967), and Cairns et al. (1968). Of these indices, only two are of concern, the Shannon-Weaver Index (1949), and the Brillouin Index (1962), because of their mathematical properties (Pielou 1975), which are nonnormal distributions, with heteroscedastic variances, and non-additive qualities.

Pielou (1975) recommended the use of the Shannon-Weaver Index when estimating the diversity of a large community and the Brillouin Index when analyzing a collection. A collection refers to a fully censused community. Pielou (1975) lists examples of collections as the breeding birds in a small tract of forest, the shrubs and trees that have established themselves in an abandoned quarry, or the catch in an insect light trap. In the sampling devices used for aquatic organisms one is dealing with collections. Pielou (1975) states that the Shannon-Weaver Index is only for large communities, that is, communities that can be treated as infinite in the sense that removing a sample from them causes no perceptible change in them. Resource managers interested in assessing impacts of contaminants are dealing with collections and should use the Brillouin Index to measure diversity. Brillouin's Index is calculated using the following formula (Pielou 1975):

$$H = \frac{1}{N} \log \frac{N!}{(N_1! \times N_2! \times \ldots\ldots N_i!)}$$

where $H$ = diversity

$N$ = total number of species in sample

$N!$ = $N$ factorial

The Shannon-Weaver Index is calculated using the following formula (Pielou 1975):

$$H = -\sum_{i=1}^{s} (p_i)(\log p_i)$$

where $H$ = index of species diversity

$P_i$ = proportion of total sample belonging to the ith species

$s$ = number of species

### Similarity Indices

Similarity indices are used to compare collections for likeness, usually with a value of 1 representing two identical collections and a value of 0 representing totally different collections. Perkins (1983) stated that because they are scaled from 0 to 1, absolute changes in community similarity may be such that they can be grouped into high, medium, and low impact categories. The use of similarity indices allows comparisons between faunal and floral samples taken at different times, places, or by different techniques (Wolda 1981).

Several similarity indices have been proposed by the following authors: Renokonen (1938), Dice (1945), Sorenson (1948), Bray and Curtis (1957), Morisita (1959), Mountford (1962), Horn (1966), Lance and Williams (1967), Clifford and Stephenson (1975), Baroni-Urbani and Buser (1976), and Wolda (1978). All but one of these indices, the Morisita Index, are affected by sample size and diversity (Wolda 1981). Wolda (1981) reported that Huhta (1979) tested a long series of similarity indices using real data and found that results depend largely on the index chosen, which suggests the dangerous possibility that one can choose an index to demonstrate whatever one wants the data to show, without necessarily being able to prove that this is indeed what they do show. Therefore, Wolda (1981) recommended the Morisita Index to avoid complex dealings with sample size and diversity.

To our knowledge, the Morisita Index has not been used to study impacts of contaminants, but appears to be a promising technique. An advantage of the Morisita Index over diversity indices is that it incorporates species richness, species composition, and relative abundance (Wolda 1981). (See Table 9 for the Morisita Index formula and a sample computation.)

### Recommendations Regarding Choice of an Index

Biotic, diversity, and similarity indices may all be useful in determining impacts of contaminants; however, we recommend the use of the Morisita Index to examine changes in community composition, because it is not influenced by diversity and sample size. The Brillouin Diversity Index also is useful because one can compare values, which is a disadvantage of similarity indices. However, usually three samples are collected from each sampling station in a study, which enables the use of nonparametric tests (e.g., Mann-Whitney U test) to compare similarity indices.

In addition to indices, relative abundance graphs pictorially display community structure, providing a useful tool in examining structural relationships within communities (James and Rathbun 1981). These graphs can be useful to resource managers in communicating results.

To compare relative abundance graphs, the Morisita Similarity Index, and the Brillouin Diversity Index, two sets of data were created for analysis (Table 6). The data were created to show how diversity indices fail to reflect changes in species composition. This was done by having both samples consist of the same species, with only the abundances rearranged. A sample size of 100 was chosen to facilitate conversions to percentages for relative abundance calculations. To make the example more meaningful, one data set is labeled as a control site and the other a point-source pollution site.

Figure 1 was designed to show how the species in Table 6 change position in terms of their relative abundances between the two data sets. Notice how "A" changes from most abundant in the control to least abundant in the point-source site.

Table 6. Created data for two sampling stations representing a control site and a point-source polluted site.

| Species | Site 1<br>Control | Site 2<br>Point-source pollution |
|---|---|---|
| A | 35 | 1 |
| B | 22 | 1 |
| C | 16 | 4 |
| D | 10 | 6 |
| E | 6 | 3 |
| F | 4 | 35 |
| G | 3 | 10 |
| H | 2 | 2 |
| I | 1 | 16 |
| J | 1 | 22 |

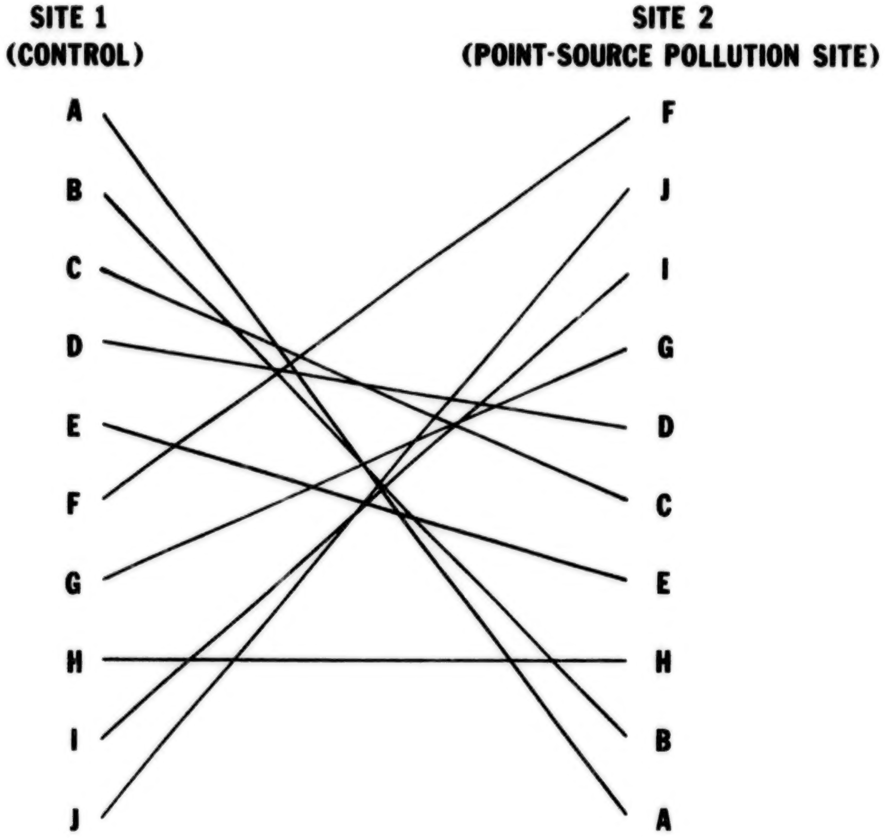

Figure 1. Display of changes in species rank between the two sites from Table 6. Notice changes in ranks, which reflect compositional changes.

Figure 2 is a plot of relative abundances for the two sites; the x-axis represents the species rank from highest to lowest and the y-axis the log of the percentage of the species in the sample. Notice how the exact same pattern exists between samples. Only the species composition changes.

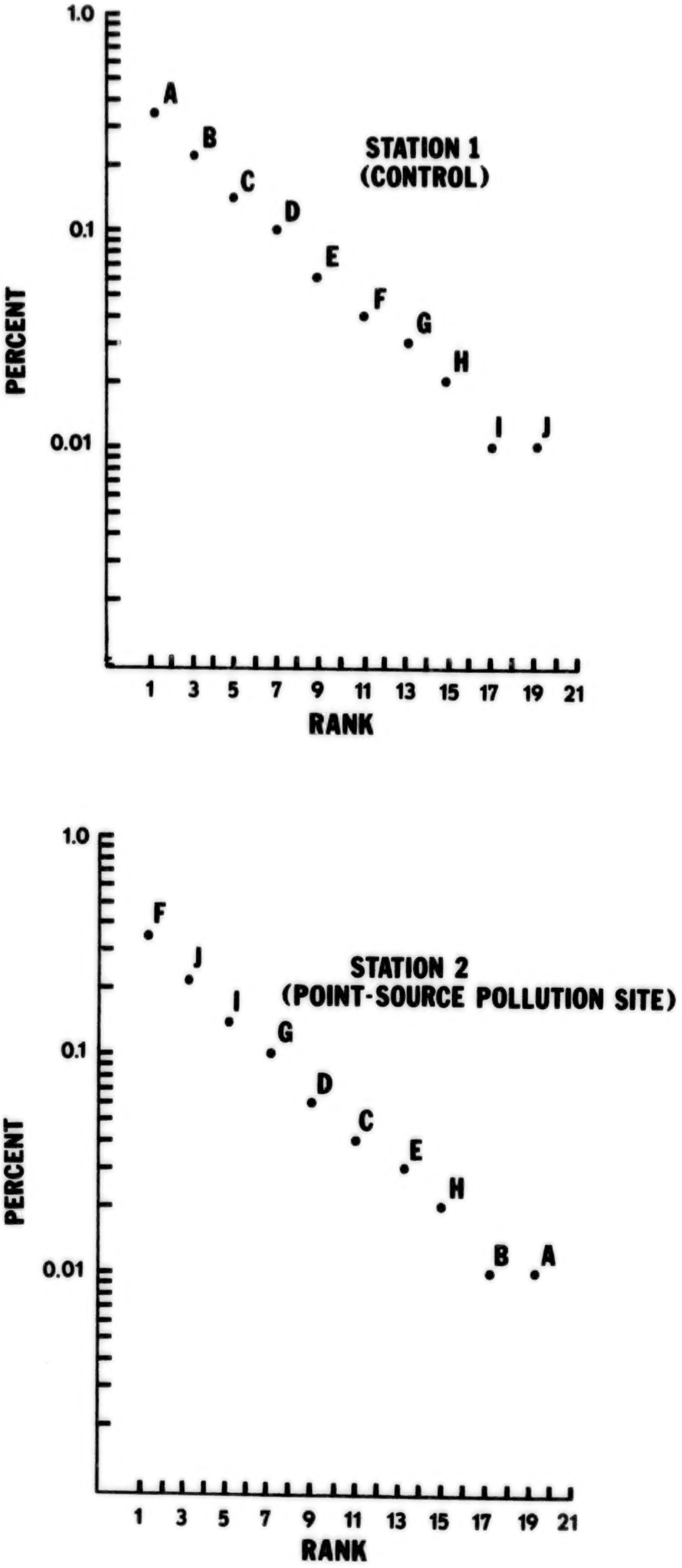

Figure 2. Relative abundance graphs of both sites from Table 6. Notice changes in relative abundances of species and thus composition.

Calculations of the Brillouin Index for each community result in a value of 0.71, reflecting similar diversity or community pattern (structure), which can be seen in Figure 2. However, if we consider species composition (Figure 1), the two samples are quite different. This difference in species composition is reflected in the Morisita Similarity Index, which was 0.20 for the two samples. The Morisita Index is scaled from 0 to 1, with 1 representing total similarity and 0 representing total dissimilarity. Based on this scale, the value of 0.20 represents samples that are more dissimilar than similar. We could conclude that the species composition between samples changed dramatically, while diversity remained constant. Thus, it appears that for this example a contaminant effect can be documented by changes in species composition.

One note of warning, the above example was designed to convey concepts of how the Morisita Similarity Index and Brillouin Index are used in determining impacts of contaminants. In an actual field investigation replicate samples should be taken to assess the variability of the system.

## APPLYING TECHNIQUES: A RESOURCE MANAGER'S RESPONSE TO A TOXIC SPILL

Suppose a tanker truck traveling through Casper, Wyoming, carrying a load of toxic chemical X runs off the road and crashes on the bank of the North Platte River. As a resource manager sitting in your office you overhear the news on the radio. You realize the spill has occurred near one of your management areas. What should you do? The following is a list of considerations for a plan of action. This scenario only refers to macroinvertebrates, although the resource manager also may decide to collect data on fish responses (American Fisheries Society 1982).

1. Contact the local fire department to acquire information concerning the type of chemical spilled, because it responds at the local level and has on-site authority. If the spill is determined to be of reported quantity, the Environmental Protection Agency is notified. Regardless, the fire department should have the best information available and could tell you chemical X is the contaminant of concern.

2. Now that you know what contaminant you are dealing with, you can call the U.S. Environmental Protection Agency to get more information regarding the properties of the contaminant. For example, you may find that chemical X dissolves in water or adheres to sediment.

3. If money is available, collect water samples upstream from the spill, at the spill, and at several stations downstream to determine concentrations of chemical X. Sampling should be carried out as frequently as possible to determine the extent and duration of the spill. You may wish to sample the day of the spill and a few days later. This depends largely on the size and chemistry of the spill.

4. Collection of flow data from gauging stations or actual field stations is important to determine the dilution of contaminant X by the river. If the spill occurs during spring runoff, damage would be expected to be much lower than at low flow stages.

5. The sediment load of the river also should be considered, because soil particles often bind with chemicals to render them harmless (Sprague 1985).

6. If lab space is available, toxicity tests may be extremely useful. These tests use water collected on-site. Fathead minnows (Pimephales promelas) and Ceriodaphnia sp. are test species that give good information regarding contaminant effects on growth and development (Mount et al. 1984). The advantages of such tests are that they document a contaminant effect on sensitive species and serve as a good indicator as to what is going on in the field (Norberg and Mount 1985).

7. Sample the macroinvertebrates at stations upstream, at, and downstream from the spill to determine if these communities have been impacted. Remember to sample riffles, pools, and intermediate zones at each sampling station to get a representative sample of the communities. Drift nets may also be useful in assessing macroinvertebrate response.

8. Continue sampling after the spill has been cleaned up, to monitor long term effects and recovery of the site.

The above eight steps may be cost prohibitive, but they provide options for assessing the impact of contaminant X. Realistically, a resource manager may be able to acquire information only from the fire department to assess the impact on the resource. Ideally, models could be developed to measure a minimal number of variables to predict the possible impact of the spill. One possibility would be to inventory sites and determine sensitive species, much like the tolerance factors of biotic indices. Further work on the development and testing of such models could prove to be valuable.

Assume that the resource manager in charge of this spill chose to use macroinvertebrates to assess the impact of the contaminant, and that he sampled immediately following the spill. Figure 3 shows a map of the spill site and the sampling stations selected by the manager. Notice that the manager was careful in selecting sites: site 1 is located upstream to serve as a control, site 2 is at the location of the spill, site 3 is downstream from the spill, and site 4 is at a location still farther downstream and slightly downstream from a major tributary (Figure 3). The macroinvertebrate data collected from the four stations are given in Table 7.

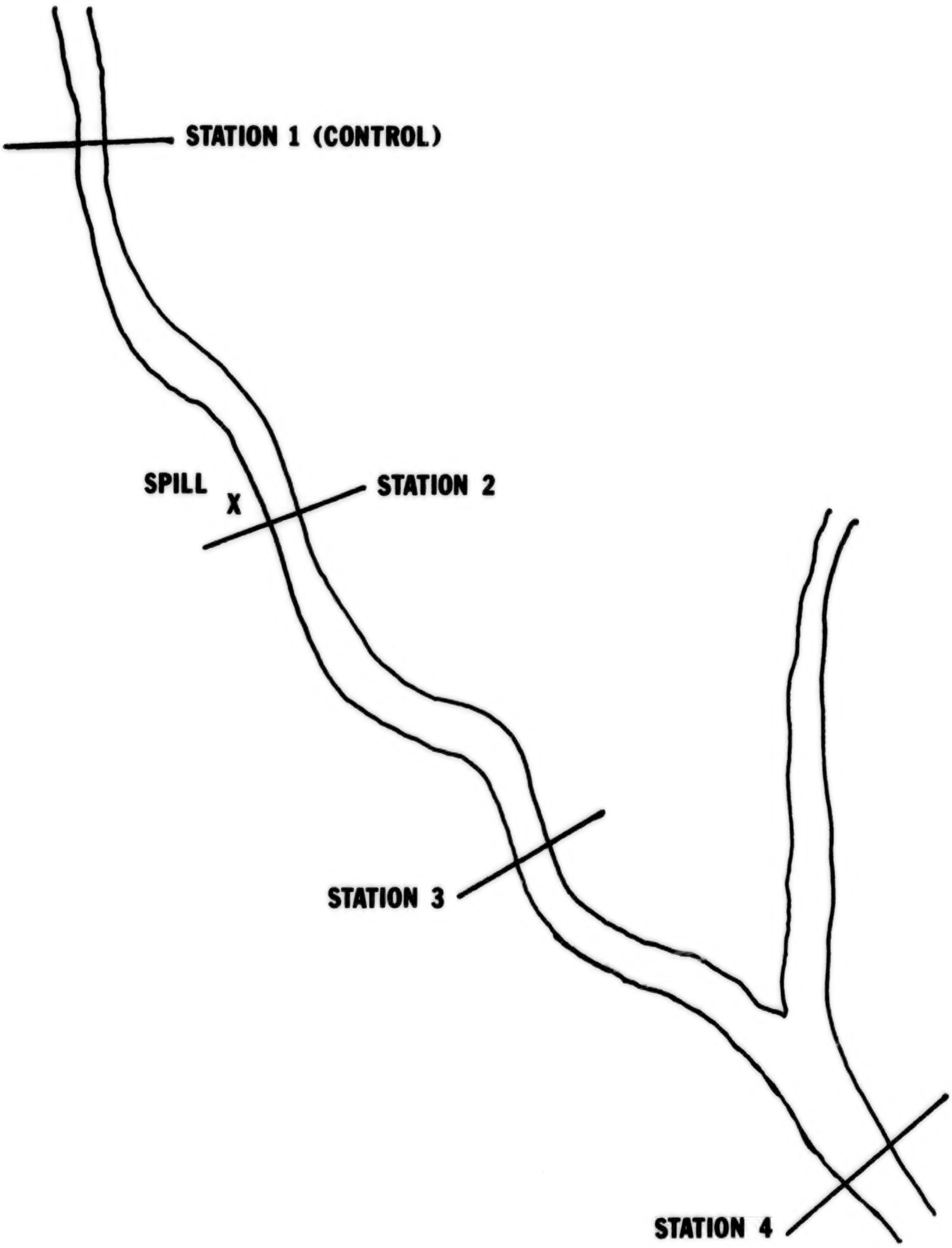

Figure 3. Map of spill site and sampling stations selected by the resource manager.

Table 7. Number and percent of invertebrates species collected at the four sampling stations. Numbers in parentheses are the percentages of each species in the sample for use in construction of relative abundance graphs displayed in Figure 4.

| Taxonomic group | Sampling station | | | |
|---|---|---|---|---|
| | 1 | 2 | 3 | 4 |
| Oligochaetes | 3 (.07) | 20 (.27) | 10 (.19) | 15 (.33) |
| Stoneflies | 5 (.11) | 0 | 1 (.02) | 4 (.09) |
| Mayflies | 3 (.07) | 0 | 1 (.02) | 2 (.04) |
| Megalops | 1 (.02) | 0 | 0 | 1 (.02) |
| Beetles | 4 (.09) | 1 (.01) | 1 (.02) | 3 (.07) |
| Caddisflies | 10 (.23) | 3 (.04) | 5 (.10) | 9 (.20) |
| Midges | 10 (.23) | 50 (.68) | 30 (.58) | 15 (.33) |
| Other dipterans | 8 (.18) | 0 | 4 (.08) | 6 (.13) |

Figure 4 displays the relative abundance graphs for each of the four sampling stations in Table 7. One of the important features of a relative abundance graph is the species found at low percentages on the right side of the plot. These species represent rare forms that may be the first to disappear in the event of a perturbation. This is evident by the loss of megalops between stations 1 and 2 (Figure 4). Also, notice how the shapes of the graphs change as one goes from the control, to the spill site, to the downstream site, and finally to the site below the tributary. These changes in shapes show how the river macroinvertebrates are initially impacted by the spill and how recovery occurs further downstream.

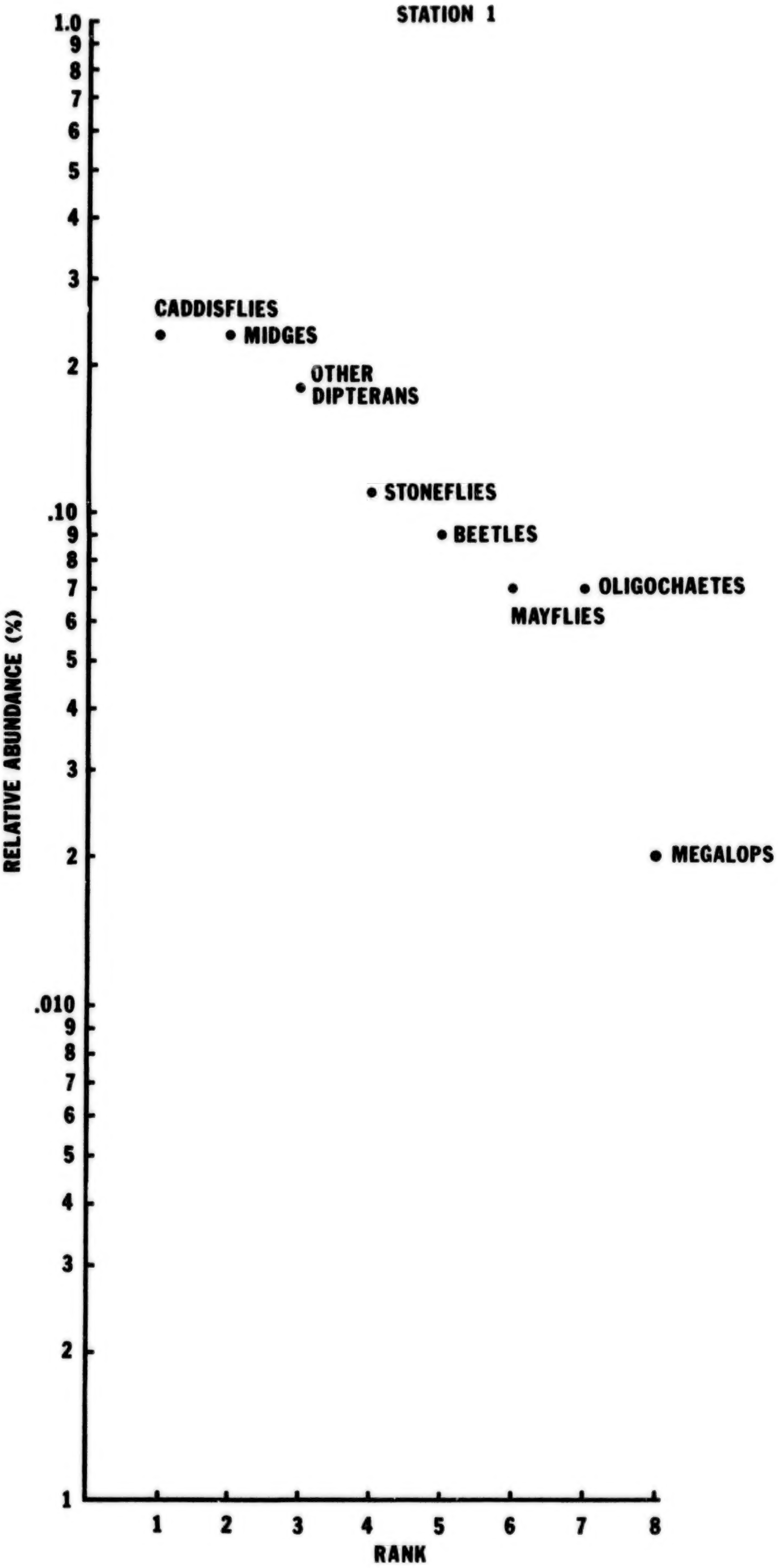

Figure 4. Relative abundance and rank information for organisms.

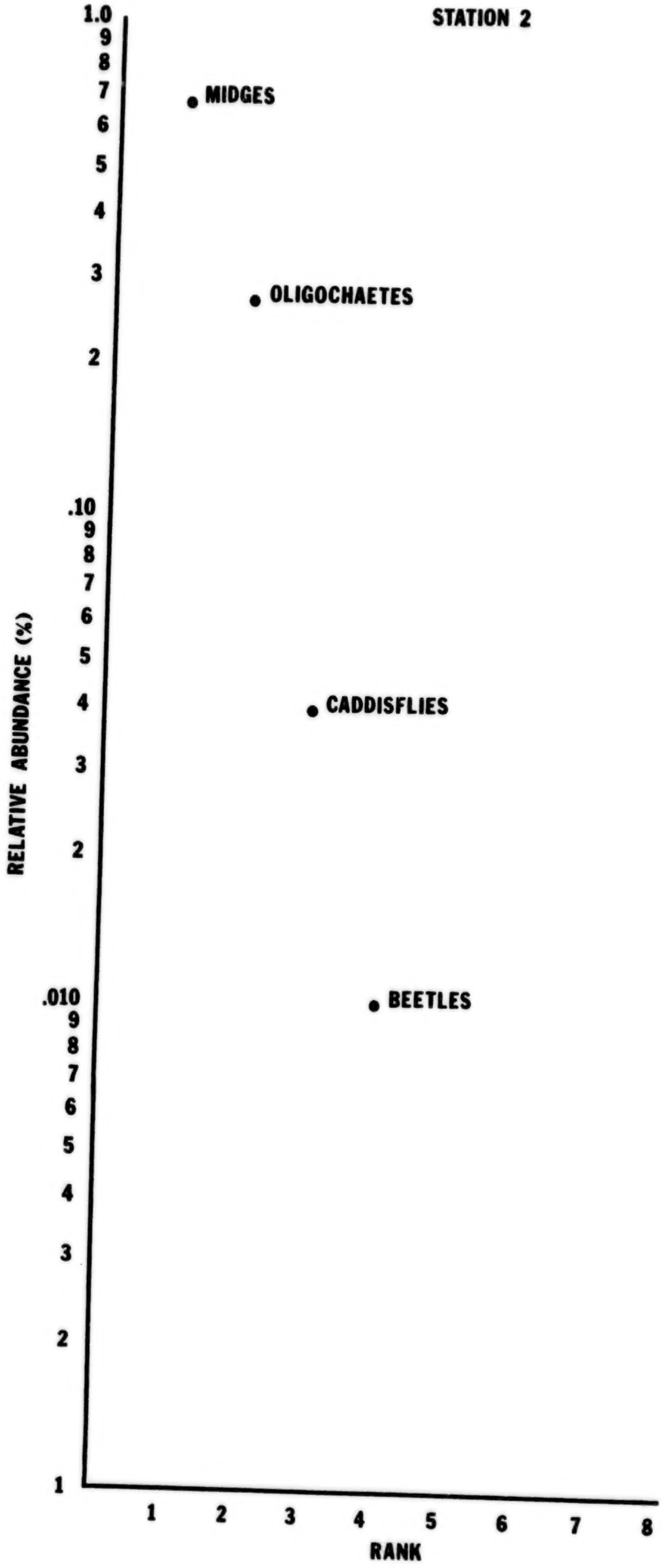

Figure 4. (Continued)

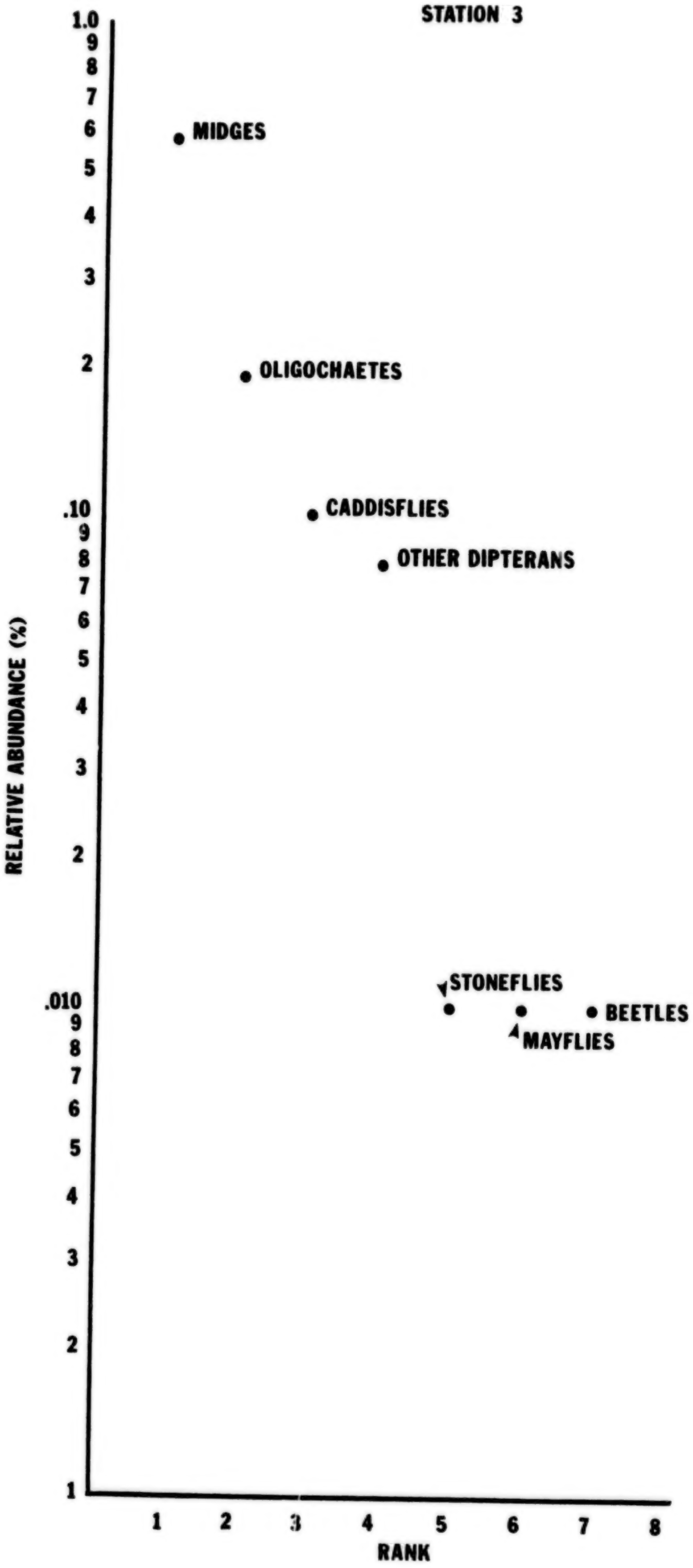

Figure 4. (Continued)

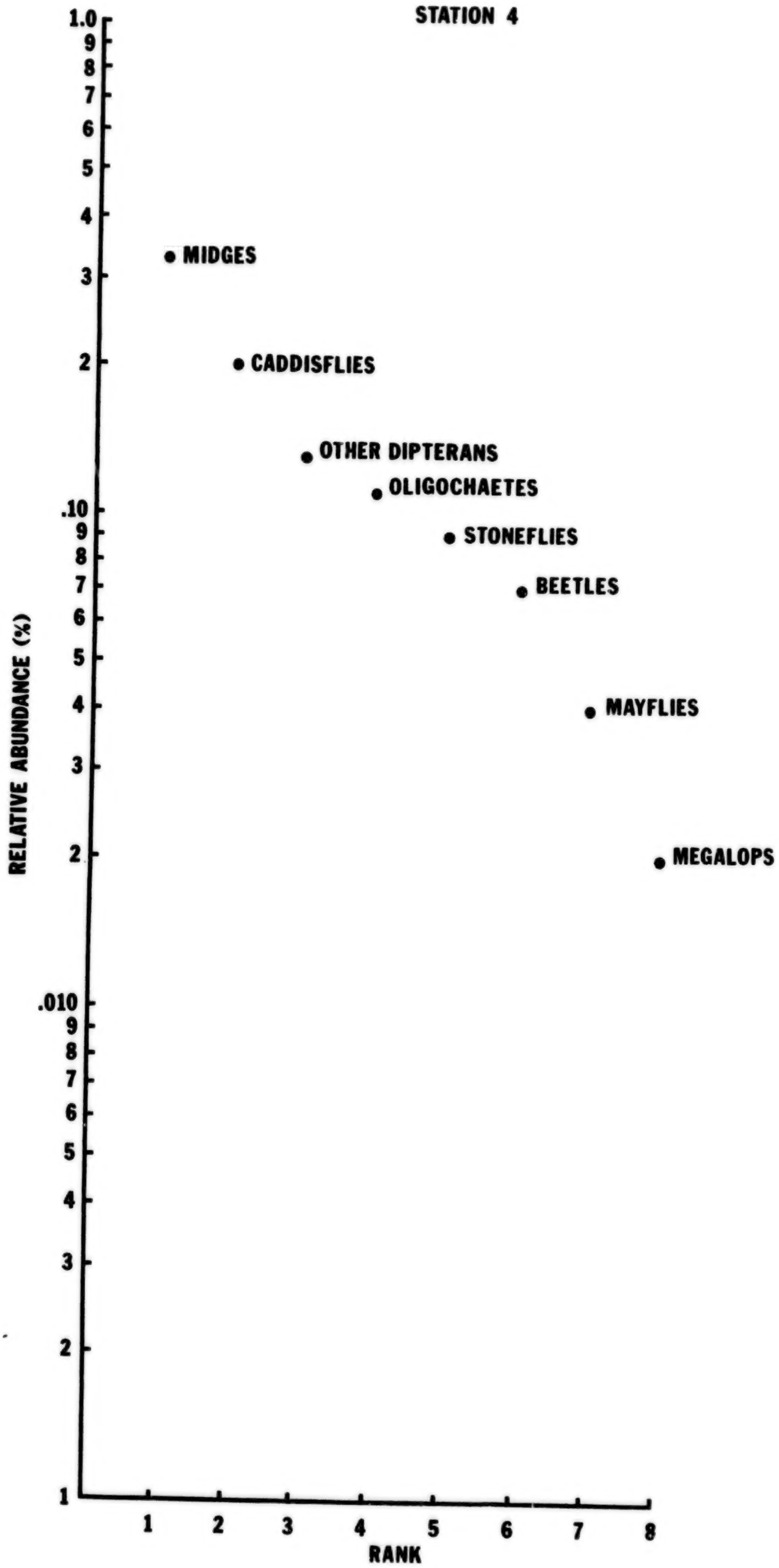

Figure 4. (Concluded)

Table 8 displays Brillouin Diversity Index values for each station, with diversity decreasing at the spill and increasing further downstream to approach that of the control.

Table 8. Brillouin Diversity Index values for the four sampling stations.

| | Sampling stations | | | |
|---|---|---|---|---|
| | 1 | 2 | 3 | 4 |
| Diversity value | 0.72 | 0.32 | 0.49 | 0.69 |

For example, Sampling Station 1 (Table 7)

$$H = \frac{1}{N} \log \frac{N!}{(N_1! \times N_2! \times \ldots \times N_1!)}$$

$$H = \frac{1}{44} \log \frac{44!}{3! \times 5! \times 3! \times 1! \times 4! \times 10! \times 10! \times 8!}$$

$$H = \frac{1}{44} \log 4.829 \times 10^{31}$$

$$H = 0.72$$

Table 9 displays Morisita Similarity Index values for between-site comparisons in a correlation table format. Table 9 demonstrates that sites 1 and 4, 2 and 3, and 3 and 4 are highly similar, with values of similarity equal to 1.1, 0.99, and 0.89, respectively. These high similarities can be interpreted to mean that the river has recovered by the time it has reached station 4, which is showing signs of both recovery and damage. The value of 0.55 for stations 1 and 2 indicates that the aquatic macroinvertebrate community has been impacted, which also is supported by the diversity data. While an impact has been documented, both the similarity and diversity data indicate that at station 4, further downstream, no measurable impact was noted.

Table 9. Morisita Similarity Index values for the four sampling stations.

| Sampling stations | Sampling stations<br>1 | 2 | 3 | 4 |
|---|---|---|---|---|
| 1 | - | | | |
| 2 | 0.55 | - | | |
| 3 | 0.73 | 0.99 | | |
| 4 | 1.10 | 0.75 | 0.89 | - |

Morisita Similarity Index (Wolda 1981):

$$C_\lambda = \frac{2\Sigma n_{1i}\ n_{2i}}{(\lambda_1 + \lambda_2)N_1 N_2}\ ,\ \text{where}\ \lambda_j = \frac{n_{ji}(n_{ji})}{N_j(N_j - 1)}$$

$n_{ji}$ = the number of individuals of species i in sample j

$N_j$ = the number of individuals in sample j

$C_\lambda$ = Morisita Index

$$\lambda_1 = \frac{6 + 20 + 6 + 12 + 90 + 90 + 56}{(44)(43)} \quad \frac{280}{1{,}892} = 0.14799$$

$$\lambda_2 = \frac{380 + 6 + 2{,}450}{(74)(73)} \quad \frac{2{,}386}{5{,}402} = 0.52499$$

$$C_\lambda = \frac{2(60 + 4 + 30 + 500)}{(.14799 + .52499)\ 44 \times 74} = \frac{1{,}188}{2{,}191}$$

$$C_\lambda = 0.55$$

Note: To simplify this example, the number of species for each order on Table 7 were considered the number of individuals for that species. For example, there were three individuals for the species Oligochaetes. Normally, the biologist would have three species in the order Oligochaetes and the number of individuals of each species.

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# APPENDIX

## Further references regarding taxonomic keys, sampling techniques, and general information regarding aquatic organisms

If one is looking for a simple book to serve as a reference for the use of aquatic organisms to monitor water quality, he should obtain a copy of Standard Methods for the Examination of Water and Wastewater, prepared and published jointly by the American Public Health Association, American Water Works Association, and the Water Pollution Control Federation. The most current edition is number 15, which was published in 1980. Othe references that could prove useful are given under the following subheadings:

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This report contains a review of literature regarding use of aquatic organisms to assess water pollution, different methods that can be used to use aquatic organisms to evaluate impacts, and an example of how data can be collected and analyzed for a field problem. The material is designed for managers and field biologists unfamiliar with aquatic organisms and their use in impact analyses. The material is written primarily for applications to contaminant-related problems in aquatic ecosystems.

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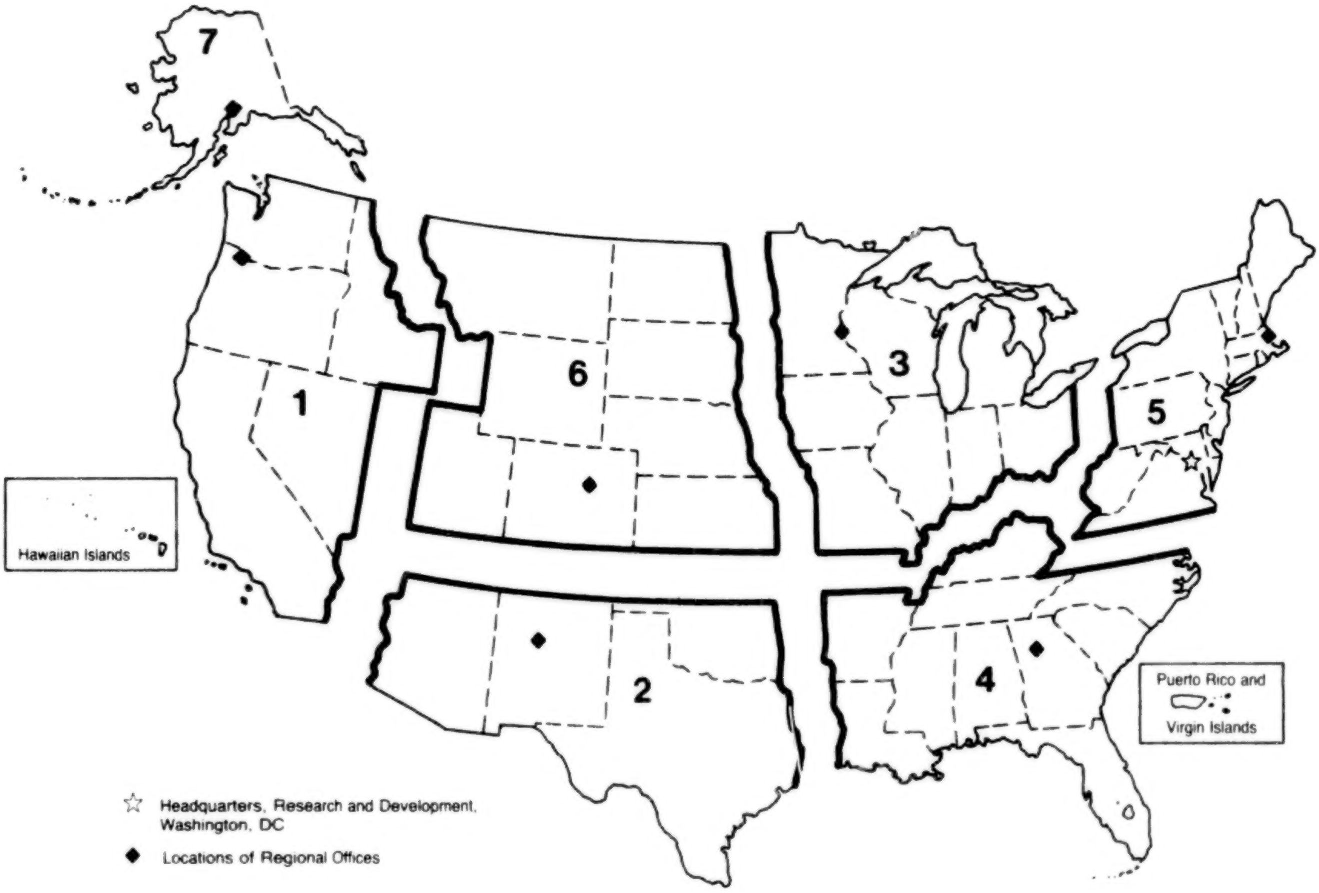

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**DEPARTMENT OF THE INTERIOR**
**U.S. FISH AND WILDLIFE SERVICE**

As the Nation's principal conservation agency, the Department of the Interior has responsibility for most of our nationally owned public lands and natural resources. This includes fostering the wisest use of our land and water resources, protecting our fish and wildlife, preserving the environmental and cultural values of our national parks and historical places, and providing for the enjoyment of life through outdoor recreation. The Department assesses our energy and mineral resources and works to assure that their development is in the best interests of all our people. The Department also has a major responsibility for American Indian reservation communities and for people who live in island territories under U.S. administration.

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